Renewable and Sustainable Energy Reviews 16 (2012) 329–341

Contents lists available at SciVerse ScienceDirect

Renewable and Sustainable Energy Reviews

journal homepage: www.elsevier.com/locate/rser

A review of the potential water quality impacts of tidal renewable energy systems

Margaret Kadiri*, Reza Ahmadian, Bettina Bockelmann-Evans, William Rauen, Roger Falconer

Hydro-environmental Research Centre, School of Engineering, Cardiff University, The Parade, Cardiff CF24 3AA, UK

ARTICLE INFO

Article history:
Received 24 May 2011
Accepted 5 July 2011
Available online 15 September 2011

Keywords:
Renewable
Barrage
Stream turbine
Tidal
Water quality
Electricity

ABSTRACT

The continued increase in the demand for energy, growing recognition of climate change impacts, high oil and gas prices and the rapid depletion of fossil fuel reserves have led to an increased interest in the mass generation of electricity from renewable sources. Traditionally, this has been pursed through riverine hydropower plants, with onshore wind systems growing steadily in popularity and importance over the years. Other renewable energy resources, which were previously not economically attractive or technically feasible for large scale exploitation, are now being considered to form a significant part of the energy mix. Amongst these, marine and in particular, tidal energy resource has become a serious candidate for undergoing mass exploitation in the near future, particularly in places with a tidal range of 4 m or more. Tidal renewable energy systems are designed to extract the kinetic or potential energy flow and convert it into electricity. This can be achieved by placing tidal stream turbines in the path of high speed tidal currents or through tidal range schemes, where low head turbines are encapsulated in impoundment structures, much like in low head riverine hydropower schemes. It is thought that these systems, when implemented at scales required to generate substantial amounts of electricity, have the potential to significantly alter the tidal flow characteristics, which could have knock-on impacts on the hydro-environment. This review gathers together knowledge from different research areas to facilitate an evaluation of the potential hydro-environmental impacts of tidal renewable energy systems, with a particular focus on water quality. It highlights the relevance of hydro-environmental modelling in assessing potential impacts of proposed schemes and identifies areas where further research is needed. A case study is presented of recent modelling studies undertaken for the Severn Estuary.

Contents

1. Introduction 330
2. Key features of tidal renewable energy systems 330
   2.1. Tidal range structures 330
   2.2. Tidal stream systems 331
3. Water quality impacts 331
   3.1. Sediment transport 332
   3.2. Salinity 332
   3.3. Dissolved oxygen (DO) 333
   3.4. Metals 333
   3.5. Nutrients 334
   3.6. Pathogens 335
4. Water quality modelling 335
5. UK case studies: hydro-environmental modelling of the impacts of tidal renewable energy systems 335
6. Conclusion 340
   References 340

* Corresponding author. Tel.: +44 2920876814; fax: +44 2920874939.
E-mail address: kadirimo@cf.ac.uk (M. Kadiri).

1364-0321/$ – see front matter
doi:10.1016/j.rser.2011.07.160

1. Introduction

In the past five decades, there has been a constant global increase in the demand for energy. According to the projections of the International Energy Agency (IEA), the global energy demand has risen by about 40% since 1990, and a 53% increase is expected by 2030 [1]. The majority of the global energy demand is still met by fossil fuels, which accounts for 88.1% (i.e. crude oil accounts for 34.8%, natural gas for 24.1% and coal for 29.2%) [2]. Due to this widespread use of fossil fuels, global reserves are depleting. The projected years of supply remaining for the global reserves of crude oil, natural gas and coal are 47, 60 and 131 years, respectively [3]. As a result, fossil fuel prices have been increasing and they are expected to continue to rise, as buyers compete for finite reserves. Furthermore, fossil fuels significantly contribute to the emission of green house gases, including carbon dioxide, which is widely regarded as one of the causes of global warming and climate change. With the global increase in the demand for energy, the growing awareness and concerns over high crude oil and natural gas prices, high carbon footprint, climate change and the rapid depletion of fossil fuel reserves, the generation of energy from renewable resources has become imperative. In addition, being a member state of the European Union, the UK is committed to the EU renewable energy target of producing 15% of its energy from renewable resources by 2020, which corresponds to 35% of the UK's electricity demand [4,5]. However, only 5% of Britain's electricity presently comes from renewable resources [6].

The total tidal energy resource in the world has been estimated and key areas mapped out for some time [7]. Practical estimates suggest that tidal energy systems can provide a significant proportion of the UK's electricity demand, reliably and safely [8]. As a result, there has been an increased interest in the UK to harness the tidal resource in recent years, with tidal range and tidal stream resource sites with potential for large scale commercial exploitation having been identified [9]. There are strong indications that the growing interest in tidal renewable energy systems will lead to the implementation of several additional schemes in the next decade or two around the UK. It is understood that such schemes can have a range of near-field and far-field impacts on the hydro-environment, which can be perceived as both positive and negative. These impacts can occur mainly through alterations to tidal flow characteristics, with changes to the hydrodynamic regime potentially having knock-on impacts on sediment transport, geomorphological, ecological and water quality processes in an estuary and/or coastline [10–12].

In this paper, we have pulled together evidence from different areas of research to evaluate the likely hydro-environmental impacts of tidal renewable energy systems, focusing on estuarine and coastal water quality. The overall objective is to inform modelling refinements and applications, as well as the formation of future research questions, ultimately to assist environmentalist and planners involved in assessing the environmental risks of tidal renewable energy systems.

2. Key features of tidal renewable energy systems

2.1. Tidal range structures

Tidal range structures create an artificial head difference between the inside and outside of a water basin and generate electricity by releasing water through low head turbines. There are two main tidal range structure designs–tidal barrages and tidal lagoons. The potential power generated by these tidal range structures can be expressed as:

$$P \propto AH^2$$

where P = potential generated power, A = wetted impounded surface area and H = head difference across the impoundment wall. Consequently, the optimal locations for tidal range structures are estuaries with high tidal ranges and large basins, many of which are present in the UK (Fig. 1) [13]. For example, the Severn Estuary, situated along the coast of South Wales and the western English county of Somerset, has the third largest tidal range in the world, with its spring tidal range reaching 14 m, demonstrating the extensive tidal power resource available within the UK. The other estuaries with tidal ranges higher than the Severn Estuary include the Bay of Fundy and Ungava Bay in Canada, each with a maximum tidal range of 17 m [14].

Tidal barrages are effectively dams built across estuaries where the tidal range is large enough to site turbines and economically generate electricity. They also act as storm surge barriers and hence, can help reduce the risk of coastal flooding. They work in a similar manner to a low-head hydropower system and extract energy from the rise and fall of the tide.

A tidal barrage usually consists of four main components [15]:

- Embankments: These are constructed where there are gaps across the estuary. They are designed not to dissipate much of the flow energy.
- Turbines: They are located in water passages and designed to convert the potential energy created by the head difference across the barrage into kinetic energy and subsequently rotational energy which is converted to electricity through generators.
- Openings: They are fitted with control gates (i.e. sluice gates) to pass flows at a particular time and with minimum obstruction.
- Locks: Usually, one or more lock(s) are built in the structure to allow ships and boats pass the barrage safely.

A number of tidal barrages have been proposed worldwide including the Penzhinskaya scheme in Russia and the Severn and Mersey schemes in the UK. However, there are few operating tidal barrages in the world. The first and largest is the La Rance Barrage in France which has been in operation since the 1960s and generates up to 240 MW. The second largest is the Annapolis Royal Barrage in

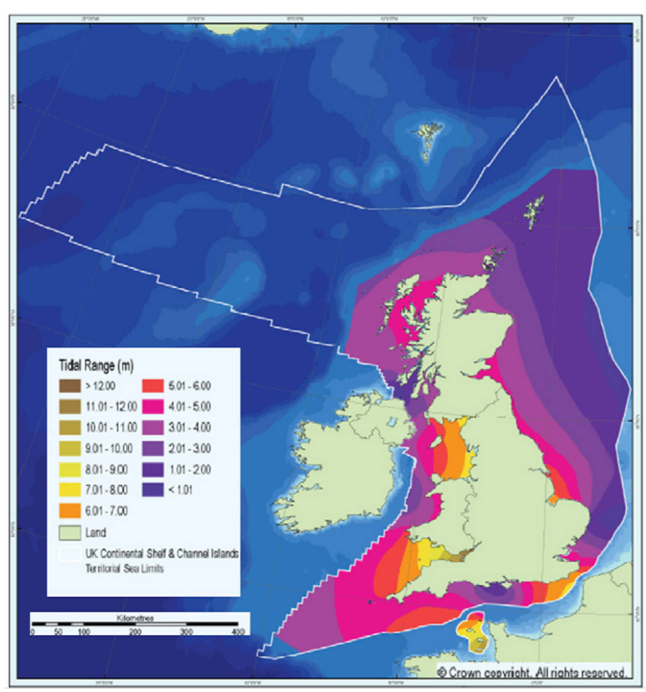

Fig. 1. Mean spring tidal range in UK estuaries [13].

Canada which generates about 18 MW [16]. Smaller schemes exist such as the Kislaya Guba plant in Russia (0.4 MW) and Jiangxia plant in China (3.2 MW) [16]. There are also schemes under construction in South Korea including the Sihwa Lake plant.

A tidal barrage can be operated with different operating schemes such as ebb generation, flood generation and ebb and flood generation (also known as two-way generation), where each of these operating schemes can be supplemented with pumping. Selecting the most effective operating scheme to maximise power output and efficiency needs to be balanced with other drivers such as navigation, flooding issues and the environmental impacts both upstream and downstream of the barrage. Between the three operating schemes, flood generation produces the least power output, ebb generation is the most appropriate for maintaining navigation and two-generation has the least impact on the hydrodynamic regime of the estuary [17]. However, ebb generation which is explained below, is the most common operating scheme, with the La Rance barrage presently operating under the ebb generation scheme. Therefore, we limit the scope of this paper to ebb generation operating scheme. Details of the other operating schemes can be found in [15]. The ebb generation scheme consists of three main stages including:

- Filling: where water flows from the sea into the basin, through both sluice gates and turbines. The water level within the basin is lower compared to the level outside the basin at this time and it will usually continue to rise until the level outside of the basin reach high-water level. The turbines do not generate electricity at this stage.
- Holding: Both turbines and sluice gates are closed at this stage. The basin water body and the sea behave as two independent water bodies. The holding stage occurs twice during each tidal cycle, namely the post-filling phase and the post-generating phase. Water levels within the basin during the post-filling phase are usually slightly lower than high water level, while water levels within the basin during the post-generating phase are usually around mean water level.
- Generating: Turbines are open at this stage and water flows from the impounded basin into the sea through the turbines. Flow is due to the head difference either side of the barrage, which is artificially generated by the holding phase, and generating electricity. The sluice gates are closed during this stage.

A tidal lagoon is an enclosed embayment which is constructed as a fully closed basin and operates under the same concept as a barrage. Tidal lagoons can be built off-shore or attached to the coast. One of the advantages of the lagoons over barrages is that they do not block the estuary and subsequently, may have a lesser impact on flow, sediment transport, fish migration and shipping than barrages. In addition, coastally attached impoundments can reduce flood risk. However, they have a higher construction cost compared to barrages [18]. This is due to the higher wall length to basin area ratio required for a tidal lagoon. Given that the Severn Estuary is one of the most attractive sites for tidal lagoons in the UK, the construction of tidal lagoons on the estuary has been proposed including the Fleming Lagoon. Other attractive sites for tidal lagoons include the estuaries on the coast of North West and South East England as well as North Wales due to their high tidal range (Fig. 1). Presently, there are no built tidal lagoons in the world.

2.2. Tidal stream systems

Tidal stream turbines are devices that extract kinetic energy from the tidal stream and convert the energy into electricity. The principle behind tidal stream turbines is similar to wind turbines. However, the working medium for stream turbines is water which is almost 1000 times denser than air. This is a more recent technology compared to barrages. Tidal stream turbines can be categorised based on their rotor configuration into horizontal axis turbines, reciprocating hydrofoil and vertical axis turbines. Generally, a horizontal axis turbine can only capture a portion of the total kinetic energy passing through the area swept by the device blades which can be represented as:

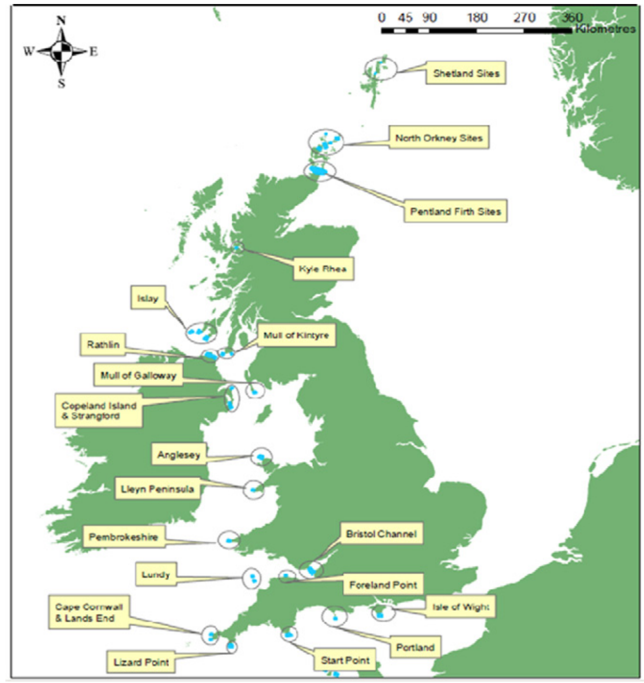

Fig. 2. Locations of tidal stream resource sites in the UK [9].

$$P_{\text{available}} = \frac{1}{2} C_p \rho A U^3$$

where C_p = power coefficient, $P_{\text{available}}$=total energy flux passing through the turbine area, ρ = water density (kg m^{-3}), A = area of the control volume (m^2) and U = component of the water flow velocity perpendicular to the cross-section of the channel (m s^{-1}). C_p is limited according to the Betz law with a theoretical maximum of 59.3% [19]. Considering the available energy for the turbines, ideal locations for stream turbines are areas with high speed currents. The maximum energy that can be extracted from the water body depends on the direction of the currents and locating the turbines in the ideal direction in the waterbody plays an important role. The Severn Estuary is one of the estuaries in the UK with the highest tidal currents, making it an ideal site for stream turbines. It is also the only tidal stream resource site in the UK with potential for large scale commercial exploitation not situated in open sea (Fig. 2) [9].

Although many studies on the design and performance of stream turbines have been conducted [20–22], there is no operational array of stream turbines to date. However, a few full scale pilot stream turbines are in operation including the SeaGen turbine installed in Strangford Lough (Northern Ireland) which generates 1.2 MW and the OpenHydro turbine installed in the Bay of Fundy (Canada) which generates 1 MW [23,24]. Several schemes are planned for installation in several locations in the UK including Ramsey Sounds in West Wales and Pentland Firth in Scotland [25].

3. Water quality impacts

The hydro-environmental parameters which impact on water quality in an estuary or a coastline as a whole are diverse,

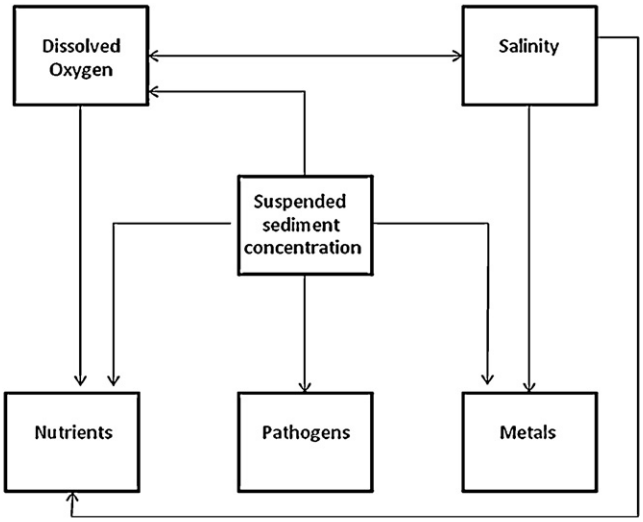

Fig. 3. Summary of inter-relationships between key water quality parameters considered in this paper.

ranging from physical to biological to chemical parameters. Therefore, this paper focuses on the key parameters which are most likely to be affected by the changes in the hydrodynamic regime following the construction of tidal renewable energy systems in macro-tidal estuaries (Fig. 3).

3.1. Sediment transport

The presence of a tidal range structure across or tidal stream device in an estuary is likely to have an effect on the pre-existing sediment transport capacity of the estuary. This would mainly be due to changes in the magnitude of tidal currents and sediment carrying capacity within the estuary [5,26]. Xia et al. [5] predicted a 50% reduction in tidal currents in the Severn Estuary following the construction of a barrage due to a decrease in the volume of water entering and leaving the estuary. Such a reduction in tidal currents is likely to lead to increased sedimentation rates in certain locations [27]. The location of potential sedimentation will depend on the prevailing source of sediment to the estuary. If riverine sources are dominant, then sedimentation is likely within the impounded water column. However, if the sediments arrive mainly from marine sources, then sedimentation is likely to occur in the vicinity immediately outside the structures [27]. In the case of tidal stream devices, the current velocities are predicted to reduce both upstream and downstream of the devices [28] and this could lead to a significant modification of sediment depositional patterns. Also, sediments that are transported back and forth within the estuary twice a day with the tides will be removed by deposition due to reduced currents. With the reduction in tidal currents, the suspended sediment levels in the water column will decrease. For example, following the construction of a barrage across the Severn Estuary, the suspended sediment levels are predicted to reduce considerably from maximum spring tide conditions of $1200\,\mathrm{mg\,l^{-1}}$ to $200\,\mathrm{mg\,l^{-1}}$ upstream of the barrage [6]. However, there may not be a significant decrease in the suspended sediment levels in the region very close to a tidal range structure or tidal stream device. This is because there is likely to be large tidal flow velocities in the immediate vicinity of a barrage or a stream turbine. For example, there would be accelerated flows in the immediate region around a single tidal stream turbine [12]. Such conditions could lead to local scour of the seabed around the structure or device [5,12]. In general, the magnitude of change in the suspended sediment levels is closely related to the condition of the local seabed geology of the estuary [5].

Since the construction of tidal range structures can lead to a decrease in the volume of water entering the estuary, the tidal force on the seabed and the bed shear stress will reduce [29]. The bed shear stress is predicted to reduce by a factor of 3–5 with the construction of a barrage across the Severn Estuary [27]. On the whole, there is likely to be a reduction in vertical mixing due to reduced tidal flows, leading to a corresponding decrease in sediment resuspension [29]. However, near the structures, mixing and sediment resuspension could be enhanced by an increased level of flow shear. Similarly, increased turbulence levels in the wake of tidal stream turbines could cause enhanced mixing and sediment resuspension.

The impact of several tidal stream turbines on suspended sediment levels is not likely to be limited to the immediate area around the turbine site and could be evident some considerable distance from the array. This is because the presence of several stream turbines in a channel could induce a change in the flow velocities, currents and flow patterns in the entire channel [12]. Ahmadian et al. [28] predicted that the introduction of an array of tidal stream turbines in the Severn Estuary would lead to a reduction in the suspended sediment levels in the water column for up to 10 km downstream from the array. However, the reduction in the suspended sediment levels was predicted to be more significant 10 km downstream from the array than within the array. Although these impacts need to be studied in more detail, there is the likelihood that the near-field effect of stream turbines on suspended sediment levels is smaller compared to the far-field effect several kilometres away from the stream turbines.

The sediment transport pathways in an estuary will be altered, possibly irreversibly, by tidal renewable energy systems and these changes could have knock-on effects on other hydro-environmental parameters in the estuary. For example, the decrease in sediment resuspension combined with the increase in sedimentation rates and the decrease in suspended sediment levels are likely to result in a decrease in the turbidity of the water column. The decrease in turbidity will, in turn, improve light penetration in the water column [26] and hence increase the depth of the photic zone. Radford and Ruardij [30] predicted an increase in the depth of the photic zone from 0.1 m to about 5 m with the construction of a barrage across the Severn Estuary. Also, the change in sediment transport and suspended sediment levels has potential implications for the fate and behaviour of sediment-associated metals, nutrients and pathogens in the water column, which are discussed in the subsequent sections.

3.2. Salinity

Similar to suspended sediments, salinity exerts an important control on several hydro-environmental parameters including metals, nutrients and dissolved oxygen with important consequences for the water quality in an estuary. Salinity influences the fate and behaviour of metals and nutrients through its influence on sorption–desorption processes [31,32]. For example, increased salinity enhances metal and nutrient desorption from sediments, increasing dissolved metal and nutrient concentrations in the water column due to increased competition from seawater cations [31,32]. Also, salinity affects dissolved oxygen concentrations through its influence on the solubility of oxygen.

Due to the reduction in the magnitude of tidal currents and the volume of water entering the estuary with the construction of a tidal range structure [26], a reduction in tidal flushing and seawater penetration within the basin is expected. This is likely to result in a decrease in salinity levels within the impounded water column [33], with the degree of change in salinity dependent on the water volume within the impounded area. However, a

possible increase in salinity has been predicted to occur following the construction of a barrage across the Severn Estuary due to increase mean water level upstream [34]. Given that the introduction of an array of stream turbines does not present any significant obstacle to tidal flushing and no significant change is expected in water levels in the Severn Estuary with the introduction of an array of stream turbines [28], there is likely to be no noticeable change in salinity levels. It is important to note that the change in the salinity levels due to the construction of tidal energy barrages is usually smaller compared to the construction of recreational and amenity barrages [35]. Nonetheless, there is a need to assess the potential impacts of proposed tidal renewable energy systems on the salinity levels in estuaries on a case by case basis, particularly through detailed modelling studies.

3.3. Dissolved oxygen (DO)

Dissolved oxygen is an important hydro-environmental parameter which is needed to sustain aquatic flora and fauna in an estuary. It affects nutrient concentrations in the water column and controls the occurrence of harmful algal blooms [35,36]. The dissolved oxygen status of an estuary is generally a balance between inputs and outputs. A deficit between the outputs and inputs of oxygen into the water column leads to very low dissolved oxygen levels or even the complete absence of dissolved oxygen in the water column which is also known as hypoxia. The input sources of oxygen into an estuary include re-aeration from the atmosphere, freshwater flows from rivers, incoming seawater, in situ photosynthetic production and chemically bound oxygen in nitrates and sulphates [35,36]. The main processes which lead to a loss of dissolved oxygen are respiration by aquatic flora and fauna as well as the microbial decomposition of detritus and sediment organic matter which exerts a strong oxygen demand given that estuarine sediments tend to be rich in organic matter [35,36]. Also, the use of oxygen by photosynthetic plants during night time results in a reduction of dissolved oxygen. The loss of dissolved oxygen is exacerbated in eutrophic waters where dead algae sink onto the sediment bed after mass algal growth events as oxygen is used in the biodegradation process. Other possible causes of low dissolved oxygen levels in an estuary include stagnation of the water column, build-up of organic matter and inputs of nutrients from storm sewers and industrial discharges, all leading to a significant biological and chemical oxygen demand [37]. An increase in temperature can also result in low dissolved oxygen levels.

There are concerns over the potential impact of tidal range structures on the dissolved oxygen levels in the water column due to the need to protect aquatic biota, including fish. The oxygen status of an estuary with the construction of a barrage is likely to be affected largely by the reduction in tidal flushing and mixing processes, which could lead to a reduction of the dissolved oxygen levels [37]. However, there is the likelihood that the decrease in turbidity and the subsequent increase in light penetration and the photic zone in the water column would increase photosynthesis, thereby increasing dissolved oxygen levels. This could be reduced, though, by photosynthetic plants which use oxygen during night time and the use of oxygen for the increased productivity of aquatic fauna. In addition, given that the solubility of oxygen is negatively correlated with salinity, any reduction in salinity levels occurring as a result of the construction of tidal renewable energy structures would enable the estuarine waters to dissolve and retain more oxygen, hence increasing the dissolved oxygen levels [33]. In the case of tidal stream turbines, they do not present a significant barrier to tidal flushing and hence, are unlikely to have any significant impact on dissolved oxygen levels in the water column.

3.4. Metals

Excessive metal concentrations in water bodies are a major environmental concern. This is mainly because of their wide spread use, potential toxic effects on aquatic biota and their long residence time in the environment as they cannot be degraded to innocuous by-products. Certain metals are essential for living organisms in trace amounts such as copper, zinc, cobalt and molybdenum but exposure to high concentrations can have toxic biological effects [38,39]. Other metals such as lead, mercury, cadmium, nickel and arsenic that have no metabolic role can also have toxic effects on organisms on exposure to high concentrations [40]. These metals can enter the food chain, resulting in bioaccumulation in higher organisms including fish and damage to human health. An example is the Minamata disease in Japan which resulted in the death of thousands of people due to mercury poisoning following the consumption of fish contaminated with methyl mercury discharged from a chemical factory into an estuary [41].

Most of the metal inputs to estuaries arise from numerous anthropogenic sources [42]. These include sewage effluents (a potential source of zinc, cadmium, lead and copper), industrial discharge (a potential source of nickel, copper, zinc and chromium), agriculture (a potential source of zinc, lead, chromium and copper), road run-off (a potential source of arsenic, zinc, cadmium, lead and nickel) and ships (a potential source of copper and zinc). The majority of metals released from these anthropogenic sources readily associates with sediments suspended in the water column. This is because sediments have a large capacity to adsorb metals which are positively charged, due to their high surface area per unit mass and high proportion of reactive coating on particle surfaces which gives the particles an overall negative surface charge [43,44]. Suspended sediments have a tendency to settle out of the water column, creating a repository for metals. However, these sediment-associated metals can be released back into the water column due to resuspension [45,46], thereby increasing their potential risk to aquatic organisms. There are several processes through which metals interact with sediment and these include adsorption and desorption through ion exchange, precipitation and dissolution of carbonate-bound metals, formation and decomposition of organic matter-metal complexes, formation and dissolution of oxyhydroxides, sorption and co-precipitation of metals by iron and manganese oxides and precipitation and dissolution of metal sulphides [47,48]. The details of these processes are complex and not the focus of this review. Our main area of concern relates to the potential impacts that tidal renewable energy systems can have on the availability of metals in the estuary.

Given that sediments suspended in the water column have a tendency to accumulate and store metals and given the likelihood that sedimentation would increase and resuspension would decrease, on average, the construction of a tidal range structure would likely reduce total metal concentrations in the water column in the region upstream of an impoundment. As suspended sediment settle out of the water column, they would trap metals into the bed sediment and the surrounding mudflats where they may remain due to a reduction in sediment disturbance by resuspension [49]. In addition, metals from existing sediment sinks are less likely to be released back into the water column due to reduced sediment resuspension, although localised net erosion may occur due to the modification of hydrodynamic forcing conditions, particularly in the vicinity of a structure. Similarly, the reduction in suspended sediment levels following the introduction of an array of stream turbines in an estuary [28] could lead to a reduction in the total metal concentrations in the water column, particularly downstream, some considerable distance away from the array. However, the acceleration of flow in the region around a turbine and the turbulent wake associated with a turbine [12] could lead to localised

sediment resuspension and a corresponding increase in metal concentrations very close to the devices. But unlike estuaries which have a tendency to act as repositories for fine sediments with high levels of metal contamination due to anthropogenic activities, the level of metal contamination in resuspended sediments in open sea tidal stream sites would probably be minimal and unlikely to directly jeopardise water quality in the region very close to the devices. The implication of the change in suspended sediment concentrations for the adsorption of new dissolved metal inputs following the construction of tidal renewable energy structures is less clear. Several studies have demonstrated that there is an inverse relationship between metal adsorption and suspended sediment concentration due to a combination of several physical and chemical mechanisms, including colloids and particle characteristics [31,44]. However, the relationship is not generally observed above certain suspended sediment concentrations [50,51]. This means, in effect, that suspended sediments may be more prone to adsorb dissolved metals in the less turbid water column following the introduction of tidal range structures or tidal stream turbines in an estuary. As a result, dissolved metal inputs may have a shorter residence time in the water column. But further investigations are needed, particularly in the context of Environmental Quality Standards in the Water Framework Directive, which considers long term averages of contaminants levels in some cases [52]. Also, a reduction in tidal flushing associated with tidal ranges structures is likely to reduce the capacity for metal inputs to be rapidly moved out from the estuary into adjacent coastal waters which may lead to a build-up of metals in the water column [53].

The change in salinity with the construction of tidal range structures is likely to have an effect on metals in the water column as salinity exerts an important control on sediment-metal interactions in estuaries through competitive and complexing reactions of seawater ions. The major seawater cations (i.e. Na^{2+}, Mg^{+}, Ca^{2+} and K^{+}) and anions (Cl^{-} and $SO_4{}^{2-}$) are effective in mobilising sediment-associated metals into the water column [54,55]. Therefore, lower salinity conditions with the construction of tidal range structures is likely to reduce the competition between seawater cations and sediment-associated metals for cation exchange binding sites on the sediment particle surfaces. This, in turn, would lead to a reduction in the displacement of metal ions into the water column and dissolved metal concentrations behind the structures where salinity would be lower. Also, there is likely to be a reduction in the formation of soluble metal complexes with the seawater anions, leading to a reduction in metal desorption from sediments into the water column. A reduction in metal concentrations in the water column will improve water quality and reduce the potential for biological uptake and toxic effects on aquatic organisms.

3.5. Nutrients

Nutrients are of significant importance in estuaries. Nutrient concentrations coupled with adequate light conditions stimulate the production of phytoplankton which forms the foundation of the estuarine food chain [56]. As a result of the availability of nutrients and high biological productivity, estuarine environments tend to provide a suitable habitat for numerous species of aquatic flora and fauna. Nutrients are essential for all living organisms as they are vital components of nucleic acids and proteins which are the building blocks for plant and animal growth. The limiting nutrients for biological productivity in freshwater and estuarine environments are phosphates and nitrates respectively [57–59]. However, the amount of evidence for consistent nitrate limitation in estuarine environments is not as firm as that for phosphate limitation in freshwater environments [58,59].

There are several sources of nutrients in an estuary and they can be autochthonous or allochthonous. The autochthonous nutrient sources include biological fixation, chemical desorption from sediment, direct precipitation, bacterial decomposition of organic matter and groundwater flow [60,61]. The main allochthonous nutrients sources are of terrestrial origin carried into the estuary by rivers and they include agricultural run-off, industrial effluents, domestic sewage, soil erosion and leachates [62]. Also, there can be direct nutrient inputs into the estuary from atmospheric deposition. On entering the estuary, nutrients are either directly assimilated by microbial populations and plant communities or become associated with sediments through ion-exchange reactions [63,64]. The sediment-associated nutrients are not readily available for biological uptake. However, the nutrient adsorption capacity of sediments is a function of the sediment composition and environmental factors such as salinity [32,65].

Excessive nutrient inputs from both the autochthonous and allochthonous sources into estuaries can cause the prolific growth of phytoplankton resulting in eutrophication. Eutrophication is defined as the nutrient enrichment of waters which leads to an array of symptomatic changes, such as increased production of algae and macrophytes, deterioration of fisheries and water quality [66]. Other changes in the water column associated with the abundance of nutrients and eutrophication include a reduction in dissolved oxygen and pH, change in turbidity and decrease in the depth of sunlight penetration [67,68]. Also, eutrophication can accelerate biological processes and, thus, induce the production of hydrogen sulphide as well as nitrous oxide, a gas which plays a significant role in the global radiation budget through the enhancement of greenhouse effect [60,69,70]. These conditions often lead to increased mortality of aquatic organisms, including invertebrates and fish populations, the reduction in biodiversity and the overall deterioration of water quality.

The availability of nutrients coupled with increased light penetration following the installation of tidal renewable energy systems is a cause for concern due to the need to safeguard estuaries from the potential effects of eutrophication on the aquatic biota and estuarine water quality. Although nutrients in suspension tend to be flushed out to sea, the restricted tidal flushing with the construction of tidal range structures may reduce nutrient dispersion and increase the residence time of nutrients in the impounded water column [71,72]. What is also less clear is whether the increased sedimentation and deposition with the installation of tidal renewable energy systems is likely to trap more nutrients and organic matter in suspension into the bed sediment, thereby reducing the nutrient levels in the water column and enhancing carbon sequestration (carbon burial) in the bed sediment. A complication, however, is that the decrease in suspended sediment levels with the introduction of tidal renewable energy systems into an estuary may reduce the capacity for nutrient inputs from rivers to be removed from the water column through adsorption to sediment [49]. A further point to consider is whether there is likely to be less nutrient inputs from chemical desorption due to the reduction in sediment resuspension. It is clear that the change in suspended sediment concentrations has some major implications for nutrient concentrations but there is a need for further investigations. A reduction in salinity with the construction of tidal range structures may increase the residence time of nutrients in sediment due to less competition between nutrients and seawater ions for ion exchange binding sites on sediment particle surfaces. Radford [73] predicted that there would be an increase in nutrient availability with the construction of a barrage in the Severn Estuary due to reduced tidal flushing and increased residence time, but nutrient concentrations would not increase considerably and eutrophication is not likely to occur. The increased nutrient availability would have positive ecological implications. Radford and Ruardij [30] predicted that the supply of nutrients combined with the increase in the average photic zone (i.e. from 0.1 m to approximately 5 m) due to reduced turbidity

with barrage construction on the Severn Estuary would lead to considerable phytoplankton growth. The increased primary production would be utilized by zooplanktons, resulting in a much greater biomass than that presently occurring in the estuary. In addition, the additional food supply provided by the increase in organic carbon from primary production is predicted to increase the population of suspension feeders and benthic deposit feeders in the estuary [30,73].

3.6. Pathogens

The main concern associated with pathogens in aquatic environments is their potential negative impact on aquatic organisms including fish and their hazard to human health which has been well documented in many studies [74,75]. Pathogens include faecal bacteria such as coliform, faecal coliform, faecal streptococci and intestinal enterococci. These organisms in aquatic environments are mainly derived from discharges of sewage and industrial effluents either directly to sea or via rivers, urban run-off from adjacent land areas, particularly those used for livestock farming, inputs from birds especially in inter-tidal areas and agriculture-related diffuse pollution [76].

Once released into the estuarine environment, pathogens either attach onto suspended sediments, particularly those with high organic matter content or exist as free-living forms in the water column [6,77]. Free living pathogens move with water flow while attached pathogens move with the suspended sediments. Only 1–30% of bacteria in natural waters are free-living, with the majority attached to sediments [6]. This is partly due to extensive urbanisation which has made natural water increasingly turbid, providing more sediment for bacteria attachment. The free-living bacteria in the water column can die-off depending on the complex interaction of sunlight intensity, nutrient deficiency, salinity and temperature with varying magnitude of importance [78]. Sunlight intensity exerts a greater control on bacteria mortality rate in the water column compared to the other factors. Bacteria associated with suspended sediments can settle-out of the water column with the sediments to create a store in the bed sediment [79]. Generally, the levels of pathogenic bacteria can be 100–2000 times higher in the bed sediment compared to the water column [6]. However, the storage in the sediment may not be permanent and the bacteria may be subject to re-release into the water column with sediment resuspension [77,79]. Therefore, changes to sediment transport would have consequences for bacteria levels in the water column in an estuary.

The reduction in suspended sediment levels with the installation of tidal renewable energy systems (up to 83% behind a barrage in the Severn Estuary [6]) is likely to have an effect on the levels of bacteria in the estuary. Ahmadian et al. [28] and Ahmadian et al. [6] predicted that there would be a decrease in bacteria levels with the introduction of an array of tidal stream turbines and the construction of a barrage in the Severn Estuary due to reduced suspended sediment levels. The reduction in suspended sediment levels would reduce inputs into the water column from bacteria adsorbed to sediments and reduce turbidity with a corresponding increase in light penetration into the water column and bacteria decay rates.

4. Water quality modelling

An appraisal of the significance of the potential hydro-environmental impacts of proposed tidal renewable energy schemes is crucial in order to ensure that the best option is implemented. This can be investigated priori with the aid of water quality models. In this way, water quality models can inform public policy decision making processes. In addition, water quality models are widely used as research tools to better understand the processes that occur within aquatic systems that can contribute to the deterioration of water quality such as eutrophication.

Water quality models can be categorised based on number of dimensions considered. The dimensions simulated by a model provide information about its complexity and suitability for specific applications. A one-dimensional (1D) model simulates the advection and dispersion of solutes and water flow in one direction (e.g. downstream), with the water body assumed to be instantaneously mixed across its width and depth. A two-dimensional (2D) model accounts for dispersion of solutes across either the width or the depth of the water body but not both. A three-dimensional (3D) model represents water flow and solute transport in all directions. These models are usually used to model water bodies with complex mixing patterns. Before modelling water quality processes in estuaries, the hydrodynamics of the estuary has to first be modelled based on a set of assumptions about flow in the estuary. Following from this, a range of physical, chemical and biological parameters are often modelled, including dissolved oxygen, nutrients, metals and phytoplanktons through solving a set of mathematical formulations [80,81].

Several water quality models including QUAL2E [82], CE-QUAL-ICM [83], DIVAST [84], MOHID [85], WASP6 [86], EFDC [87], and ELCOM-CAEDYM [88] have been developed and applied in riverine, lacustrine, estuarine and coastal environments to study water quality and ecosystem problems such as eutrophication. These models simulate the relationships between physical, chemical and biological processes in the aquatic environment. An evaluation of the processes represented in these models, particularly those associated with nutrient cycling, highlighted that only some of these models consider sediment nutrient interactions. For example, in the CAEDYM model, the process of adsorption–desorption of phosphate and ammonium to sediments was represented (Fig. 4). But in the EFDC model, only the adsorption–desorption of phosphates was represented and in the DIVAST model, the adsorption–desorption of nutrients was not considered at all (Fig. 4). In addition, the effect of environmental variables such as salinity on the coefficients used to describe the sediment-nutrient related processes in most models including CAEDYM and EFDC was not taken into account. Also, the adsorption–desorption of nutrients is represented as linear equilibrium process. However, most experimental studies indicate that non-linear processes give a better representation of the relations between nutrients and sediment [89,90]. In essence, there is room for refining the processes and coefficients used in water quality models in order that the model predictions better reflect field conditions particularly in estuarine waters. However, water quality models still provide a valuable means of evaluating the significance of potential impacts of proposed schemes on water quality and in the following section, details of such modelling studies in the UK are presented.

5. UK case studies: hydro-environmental modelling of the impacts of tidal renewable energy systems

In this section, three studies on the application of models to investigate the impacts of proposed UK tidal range structures and an array of tidal stream turbines on the hydrodynamic regime and water quality of the Severn Estuary are presented. The first study focuses on the impacts of the proposed Severn Barrage and Fleming Lagoon on hydrodynamic parameters in the Severn Estuary. The model predicts water levels and tidal currents, with and without a barrage or lagoon and the results reported here are for a mean spring tide. In this study, an unstructured triangular 2-dimensional mesh model was developed, verified and applied. Details of the model governing equations, numerical solution methods,

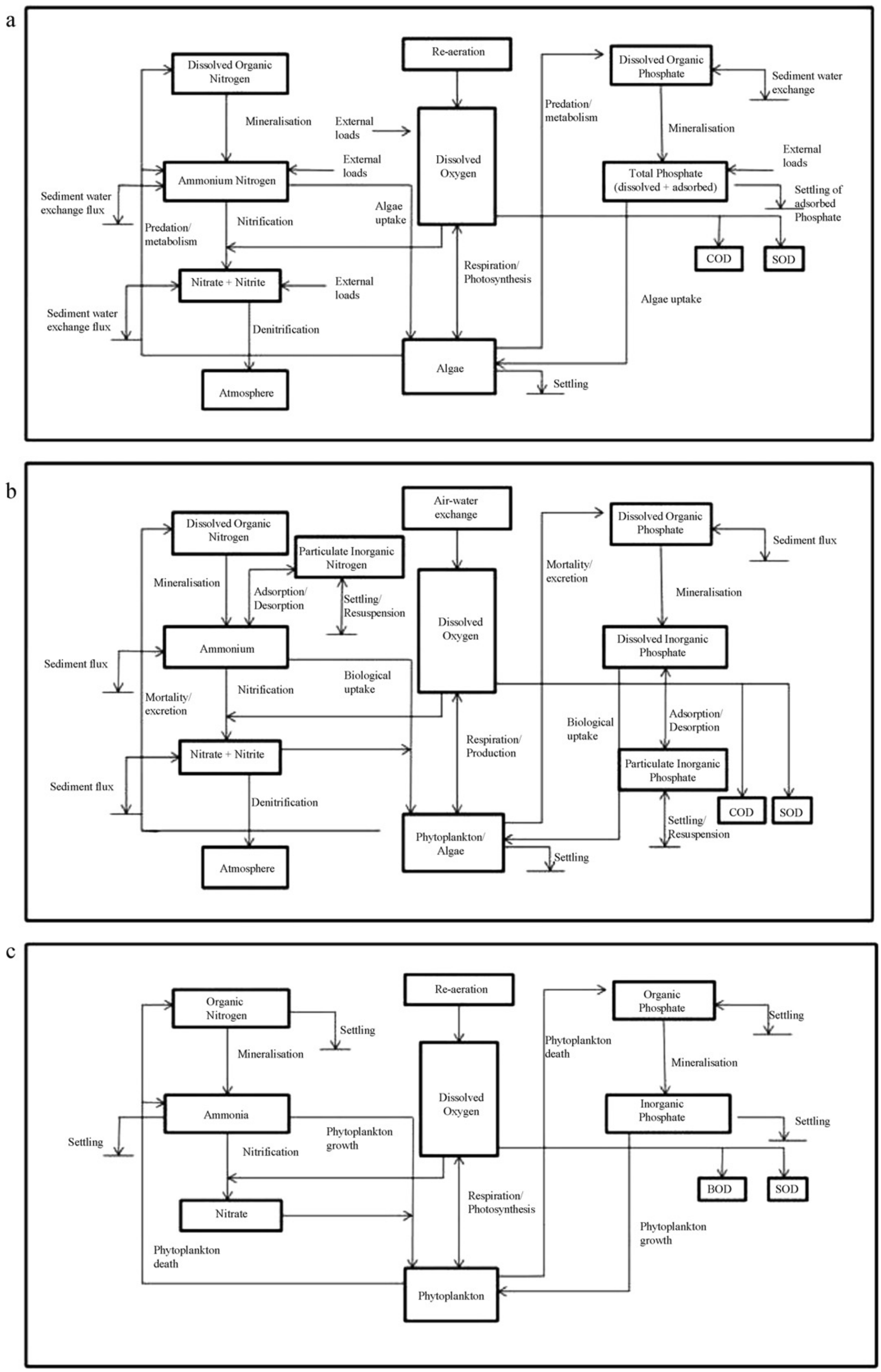

Fig. 4. Nutrient and phytoplankton cycles as illustrated in (a) EFDC, (b) CAEDYM and (c) DIVAST.

calibration and verification can be found in Falconer et al. [26] and Xia et al. [91]. The second study examined the suspended sediment concentrations and bacterial levels in the Severn Estuary as well as water level and currents, with and without the Severn Barrage. In this study, 2-D DIVAST (Depth Integrated Velocities And Solute Transport) and 1-D FASTER (Flow And Solute Transport in Estuaries and Rivers) models were validated and applied. Details of the validation can be found in Ahmadian et al. [6]. In the third study, the far-field impacts of an array of tidal stream turbines on sediment transport and bacteria levels in the Severn Estuary at mean

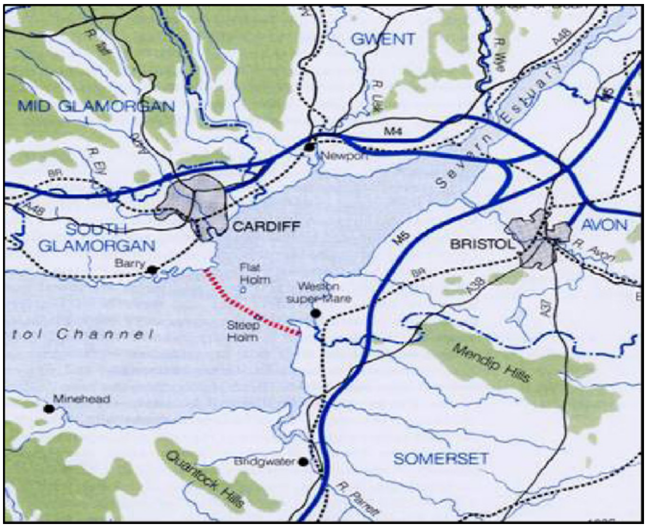

Fig. 5. Location of Severn Barrage proposed for construction in the Severn Estuary. Picture courtesy of Severn Tidal Power Group.

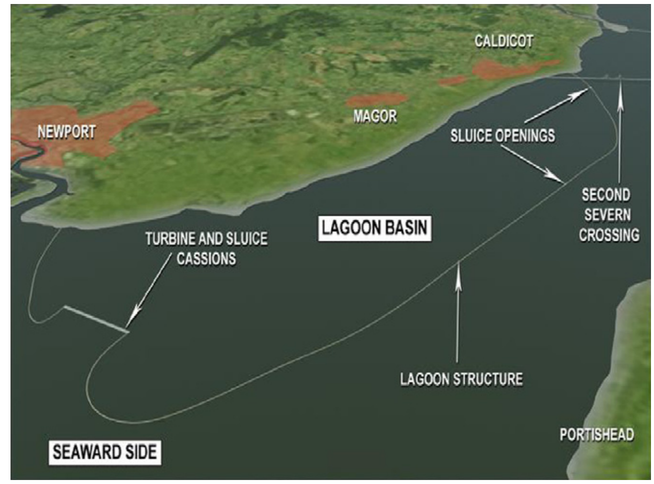

Fig. 6. Fleming coastally attached tidal lagoon – proposed for construction in the Severn Estuary.
Picture courtesy of Department of Energy and Climate Change (DECC).

ebb tide was assessed, using the 2-D DIVAST model linked to 1-D FASTER model. In this study, the model was set up for an array of 2000 m × 10 m diameter turbines located in an arbitrary site along the Severn Estuary. Details of the turbine representation and validation of the model can be found in Ahmadian [92]. The models used in these studies were developed within the Hydro-Environmental Research Centre at Cardiff University.

The Severn Estuary spans the South Wales coast and the northern reach of the South West coast of England and it is situated about 240 km west of London. The estuary is about 200 km long [26]. It has the third largest tidal range in the world [14]. The spring tidal range reaches 14 m and the spring tidal currents are well in excess of 2 m s^{-1}, demonstrating that the Severn Estuary is an ideal location for tidal range structures and tidal stream devices. There is an extensive area of intertidal mudflats along the estuary and the suspended sediment levels are high, with the spring and neap loads estimated at 30 Mt and 4 Mt respectively. As a result, light penetration through the water column is very restricted, dissolved oxygen saturation levels are low and aquatic life is limited [26].

There have been a number of barrages proposed to be built across the Severn Estuary including a barrage extending from Cardiff to Weston-Super-Mare, which is also referred to as the Severn Barrage, with a length of 16 km (Fig. 5). The barrage comprises of 166 sluice gates and 216 m × 9 m diameter bulb turbines, each producing a peak output of 40 MW, thereby peaking at 8.64 GW for a spring tide. The annual electricity generated by the barrage for an ebb-generation scheme is 17 TWh/year which is equivalent to about 5% of the electricity consumption in the UK [6,26]. Also, the barrage includes ship locks for access to ports and potentially provides flood defence as well as rail and road communications. The scheme was proposed to have an impounded area of approximately 500 km^2 and to operate under ebb tide generation conditions only (Fig. 5). Similarly, the Fleming Lagoon between Newport and the Severn Road crossing was proposed to operate under ebb generation conditions only (Fig. 6). Its impounded area of approximately 80 km^2 was proposed to have an installed capacity of approximately 1.36 GW [26].

A comparison between the model predicted water level distributions both with and without the tidal renewable energy schemes for a mean spring tide shows a reduction in the maximum water levels (Fig. 7a–c). With the construction of the Severn barrage, the maximum water levels are predicted to decrease by about 0.5 m downstream and between 0.5 and 2.0 m upstream of the barrage (Fig. 7a and b). In the case of the Fleming Lagoon, the maximum water levels following its construction are predicted to decrease by only 0.1–0.2 m in the region upstream of the lagoon (Fig. 7a and c). These predicted results indicate that there will be a greater

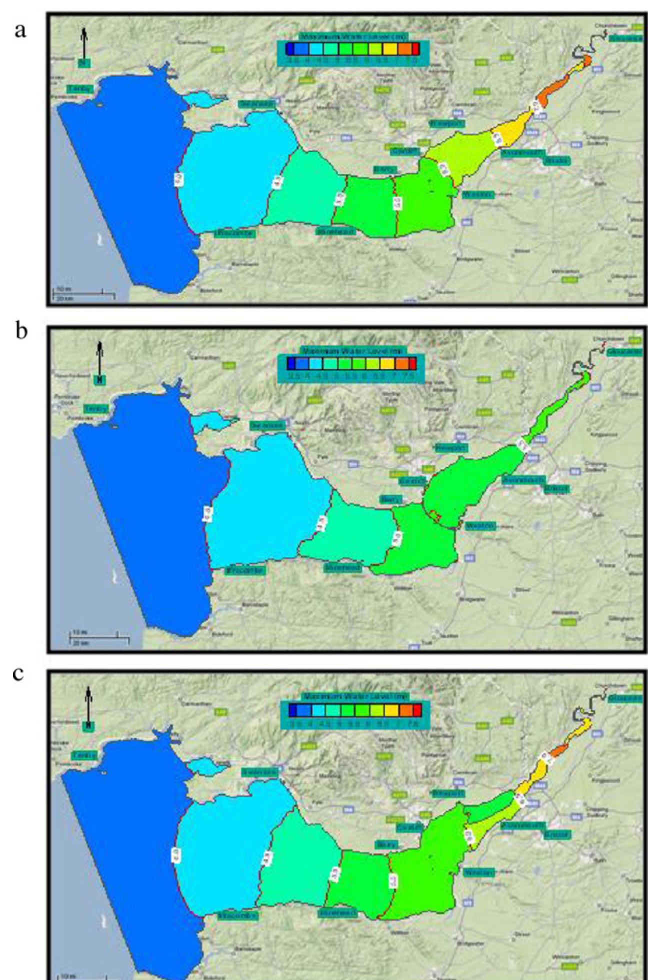

Fig. 7. Comparison of maximum spring tide water levels for (a) no scheme, (b) the Severn Barrage and (c) the Fleming Lagoon [26].

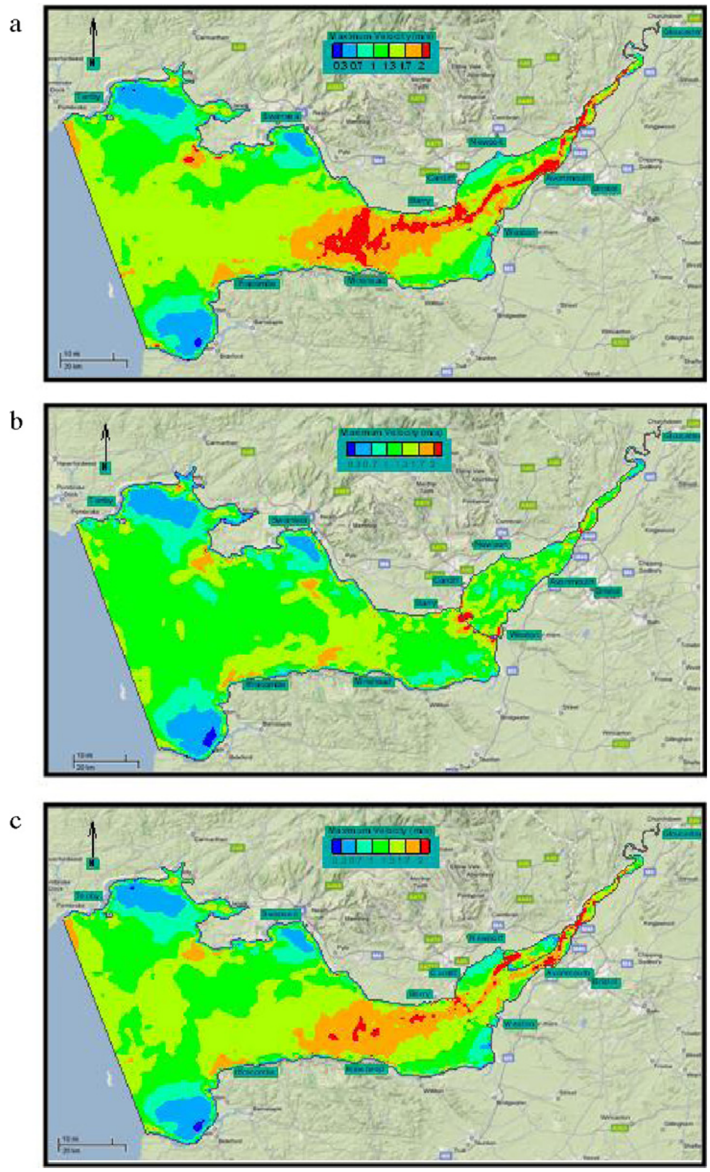

Fig. 8. Comparison of maximum spring tide currents for: (a) no scheme, (b) the Severn Barrage and (c) the Fleming Lagoon [26].

reduction in the risk of flooding upstream with the construction of the barrage compared to the lagoon.

A comparison between the predicted tidal current distributions, both without and with the Severn Barrage and the Fleming Lagoon for a mean spring tide shows a reduction in the maximum tidal currents (Fig. 8a–c). The reduction is in accordance with the decrease in water volume entering the estuary with the construction of the tidal renewable energy structures. The maximum currents with the construction of the Severn Barrage are predicted to significantly decrease in the main channel upstream of the barrage and in a fairly large area across the estuary (i.e. seawards of the barrage and west of Cardiff) (Fig. 8a and b). For example, around the outlet of the estuary, the maximum current speed is predicted to exceed $2.0\,\mathrm{m\,s^{-1}}$ without the barrage. The current speed is predicted to decrease to less than $1.4\,\mathrm{m\,s^{-1}}$ with the barrage (Fig. 8a and b). In the case of the Fleming Lagoon, the maximum currents would effectively decrease mainly upstream of the barrage line, with a slight decrease predicted in the maximum currents near the outlet of the estuary (Fig. 8a and c).

A comparison between the predicted suspended sediment levels without and with the Severn barrage for a mean spring tide shows a significant reduction with the barrage (Fig. 9a and b). For example, in the region upstream of the barrage, the suspended sediment concentration is predicted to be about $1200\,\mathrm{mg\,l^{-1}}$ without the barrage and the concentration is predicted to significantly decrease to $200\,\mathrm{mg\,l^{-1}}$ with the barrage (Fig. 9a and b). Similarly, a comparison between the predicted faecal bacteria levels without and with the barrage for mean spring tide shows a significant decrease in bacterial levels upstream with the barrage (Fig. 10a and b).

A comparison of the predicted suspended sediment levels without and with an array of tidal stream turbines for a mean ebb tide shows a reduction with the array, particularly 10 km downstream of the array (Fig. 11a and b), highlighting that the changes in suspended sediment levels is more evident some significant distance away from the array. These changes are considered to be a consequence of changes in the current velocities due to increased resistance to flow by the introduction of the stream turbines in the estuary. However, further modelling studies are required as the changes in suspended sediment levels do not follow a simple pattern during a tidal cycle. Similarly, a comparison between the predicted faecal enterococci bacteria levels without and with the array for mean ebb tide shows a decrease in bacterial levels

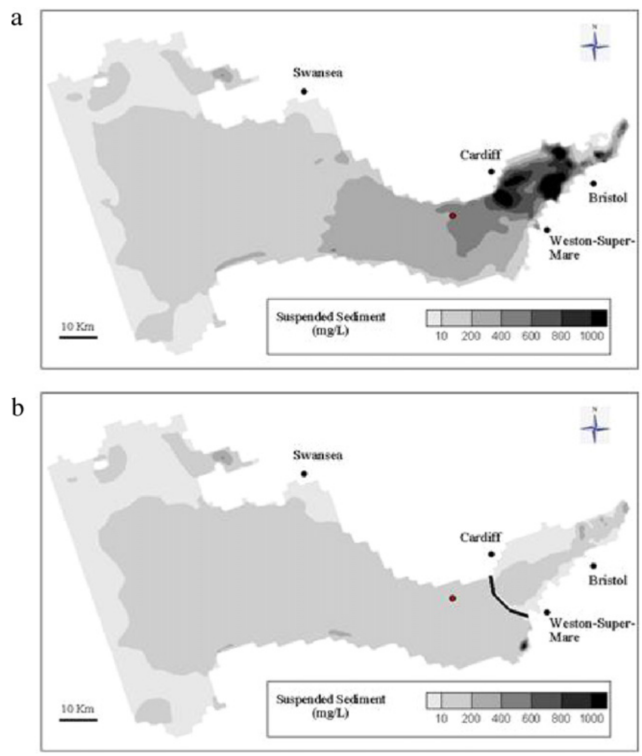

Fig. 9. Predicted suspended sediment levels across the estuary predicted at mean spring tide at Barry (red dot), (a) without and (b) with barrage [6]. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

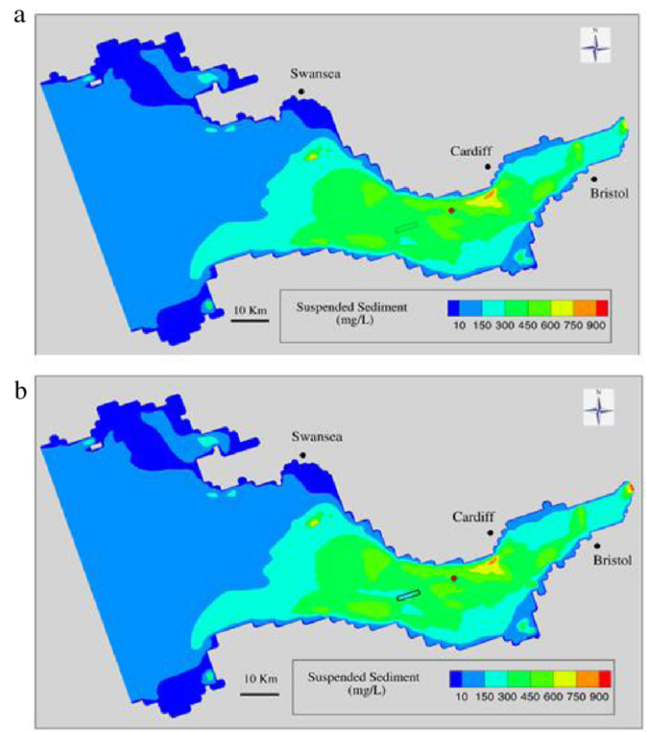

Fig. 11. Predicted suspended sediment levels across the estuary at mean ebb tide at Barry (red dot) (a) without and (b) with the turbine array [28]. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

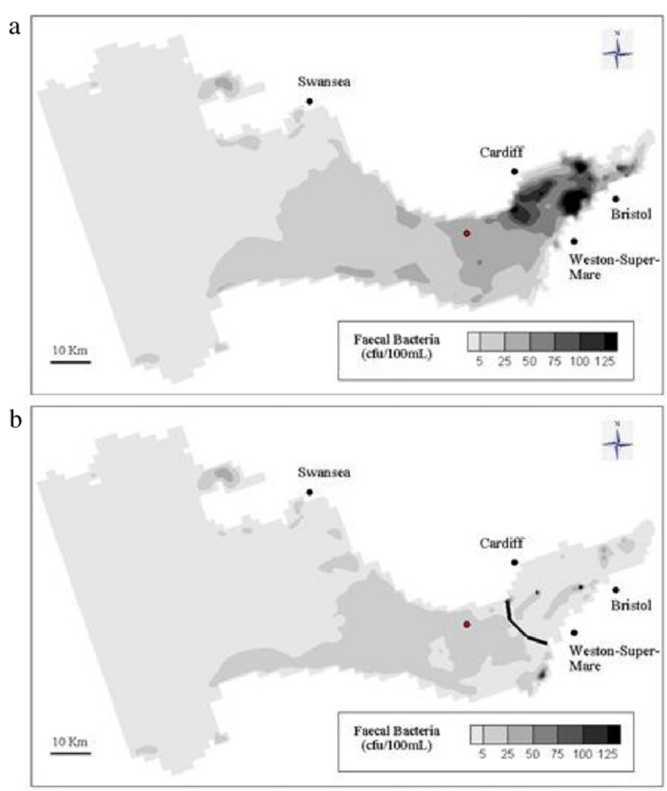

Fig. 10. Predicted faecal bacteria levels in the estuary predicted at mean spring tide at Barry (red dot), (a) without and (b) with barrage [6]. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

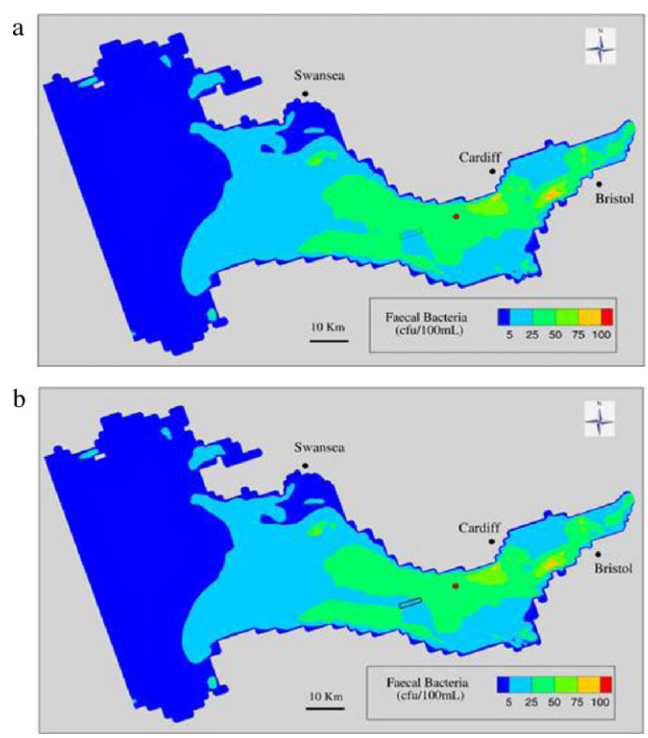

Fig. 12. Predicted faecal enterococci bacteria levels across the estuary at mean ebb tide at Barry (red dot) (a) without and (b) with the turbine array [28]. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

downstream of the array (Fig. 12a and b). However, unlike the case of the predicted suspended sediment levels, the reduction in the predicted faecal enterococci bacteria levels can be observed for up to 25 km downstream of the array. This indicates that other processes, which need to be studied in detail, contribute to bacteria transport in addition to desorption of bacteria attached to suspended sediment.

The predictions from these studies has made a significant contribution to understanding some of the changes to water quality in the Severn Estuary following the introduction of tidal renewable energy systems in the estuary. In the case of the Severn Barrage, the reduction in the magnitude of the tidal currents, would lead to a significant reduction in the suspended sediment concentrations and turbidity levels. This, in turn, would result in a corresponding increase in light penetration and a reduction in bacterial levels in the water column due to the reduced input from bacteria adsorbed onto suspended sediments and increased decay rates as a result of increased light penetration. However, further modelling studies are required to fully understand the near-field and far-field changes to suspended sediment and bacterial levels following the installation of an array of tidal stream turbines.

6. Conclusion

Globally, tidal renewable energy will play a prominent role in the generation of CO_2 free electricity in the immediate future. However, the operation of tidal renewable energy systems in or across an estuary or coastline can have significant implications for the aquatic environment. This review paper has highlighted that tidal renewable energy systems would clearly have significant hydro-environmental impacts on an estuary. Hydrodynamic and water quality modelling studies have demonstrated that the key impacts would be the change to the pre-existing hydrodynamic regime with a reduction in tidal range and currents. This would influence salinity and dissolved oxygen levels as well as sediment transport with an increase in sediment deposition and sedimentation rates. These changes would lead to a decrease in suspended sediments and faecal bacteria concentrations in the water column. The changes could also have potential knock-on effects on dissolved nutrient and metal concentrations in the water column, thereby affecting water quality, probably to a lesser extent with tidal stream devices compared to tidal range structures. Further studies are needed to fully understand the resulting effects on water quality including studies to clarify how the reduction in suspended sediment concentrations would impact nutrient availability following the installation of tidal renewable energy systems.

The significance and scale of the potential impacts of tidal renewable energy systems on water quality remains unclear and is likely to vary between estuaries and coastlines. Previous studies have shown that there are positive hydro-environmental impacts associated with tidal renewable energy systems as well as the highly publicised adverse effects. More hydro-environmental modelling studies are needed as they provide a means of assessing the significance and scale of the impacts of proposed and future schemes on a site specific basis and enable more definite predictions to be made.

References

[1] International Energy Agency (IEA). World Energy Outlook. Paris: OECD/IEA; 2008.
[2] British Petroleum Statistical Review of World Energy; 2009, June.
[3] British Petroleum Statistical Review of World Energy; 2000, June.
[4] Kirby R, Retiere C. Comparing environmental effects of Rance and Severn barrages. Proc ICE, Maritime Eng 2009;162:11–26.
[5] Xia JQ, Falconer RA, Lin B. Impact of different tidal renewable energy projects on the hydrodynamic processes in the Severn Estuary, UK. Ocean Modell 2010;32:86–104.
[6] Ahmadian R, Falconer RA, Lin B. Hydro-environmental modelling of proposed Severn barrage, UK. Proc ICE, Energy 2010;163:107–17.
[7] Charlier R. Oceans and electrical power – 2. Int J Environ Stud 1982;19:7–16.
[8] Mackay DJC. Sustainable Energy – Without the Hot Air. Cambridge: UIT Cambridge Ltd; 2009.
[9] Sustainable Development Commission (SDC). Tidal Power in the UK, Research Report 1 – UK Tidal Resource Assessment; 2007.
[10] Shaw TL. An Appraisal of Tidal Power Stations: With Particular Reference to the Severn Estuary. London: Pitman Advanced Publishing Program; 1980.
[11] Matthews ME, Young RM. Environmental impacts of small tidal power schemes. In: Clare R, editor. Tidal Power: Trends and Development, Proc. of the 4th Conference on Tidal Power. London: Thomas Telford; 1992. p. 197–214.
[12] Shields MA, Woolf DK, Grist EPM, Kerr SA, Jackson AC, Harris RE, Bell MC, Beharie R, Want A, Osalusi E, Gibb SW, Side J. Marine renewable energy: the ecological implications of altering the hydrodynamics of the marine environment. Ocean Coast Manage 2011;54:2–9.
[13] Department for Trade and Industry (DTI). Atlas of UK Marine Renewable Energy Resources: Technical Report; 2004.
[14] O'Reilly CT, Solvason R, Solomon C. Resolving the world's largest tides. In: Percy JA, Evans AJ, Wells PG, Rolston SJ, editors. The challenging Bay of Fundy – Beyond 400 years, Proc. of the 6th Bay of Fundy Workshop Report No. 23. 2004. p. 153–7.
[15] Baker AC. Tidal Power. London: Peter Perengrinus Ltd; 1991.
[16] Hammons TJ. Tidal power. Proc IEEE 1993;81:419–32.
[17] Xia J, Falconer RA, Lin BL. Impact of different operating modes for a Severn Barrage on the tidal power and flood inundation in the Severn Estuary, UK. Appl Energy 2010;87:2374–91.
[18] Baker C, Walbancke J, Leache P. Tidal Lagoon Power Generation Scheme in Swansea Bay; 2006. A report on behalf of the Department of Trade and Industry and the Welsh Development Agency.
[19] Betz A. Introduction to the Theory of Flow Machines. Oxford: Pergamon Press; 1966.
[20] Bahaj AS, Molland AF, Chaplin JR, Batten WMJ. Power and thrust measurements of marine current turbines under various hydrodynamic flow conditions in a cavitation tunnel and a towing tank. Renew Energy 2007;32: 407–26.
[21] O'Doherty T, Agaee D, Mason-Jones AA, O'Doherty D. An assessment of axial loading on a five-turbine array. Proc ICE, Energy 2009;162:57–65.
[22] Batten WMJ, Bahaj AS, Molland AF, Chaplin JR. The prediction of the hydrodynamic performance of marine current turbines. Renew Energy 2008;33:1085–96.
[23] Westwood A. SeaGen installation moves forward. Renew Energy Focus 2008;9:26–7.
[24] O'Rourke F, Boyle F, Reynolds A. Tidal current energy resource assessment in Ireland: current status and future update. Renew Sustain Energy Rev 2010;14:3206–12.
[25] Renewable UK Report, Wave and Tidal Energy in the UK-State of the Industry, 2011, March, http://www.bwea.com/ref/reports-and-studies.html (accessed 28.03.11).
[26] Falconer RA, Xia J, Lin B, Ahmadian R. The Severn Barrage and other tidal energy options: hydrodynamic and power output modelling. Sci China Ser E: Tech Sci 2009;52:3413–24.
[27] ETSU. Severn Barrage Project Detailed Report. Volume I: Tidal Hydrodynamics, Sediments, Water Quality, Land Drainage and Sea Defences; 1989.
[28] R. Ahmadian, R. Falconer, B. Bockelmann-Evans, Far-field modelling of the hydro-environmental impact of tidal stream turbines, Renew. Energy, in press, doi:10.1016/j.renene.2011.07.005.
[29] Wolf J, Walkington IA, Holt J, Burrows R. Environmental impacts of tidal power schemes. Proc ICE, Maritime Eng 2009;162:165–77.
[30] Radford PJ, Raurdij P. The validation of ecosystem models of turbid estuaries. Cont Shelf Res 1987;7:1483–7.
[31] Turner A. Trace-metal partitioning in estuaries: importance of salinity and particle concentration. Mar Chem 1996;54:27–39.
[32] Rysgaard S, Thastum P, Dalsgaard T, Christensen PB, Sloth NP. Effects of salinity on ammonium adsorption capacity, nitrification and denitrification in Danish estuarine sediments. Estuaries 1999;22:21–30.
[33] Sustainable Development Commission (SDC). Turning the Tide: Tidal Power in the UK, Technical Report; 2007.
[34] Kirby R. Distribution, transport and exchanges of fine sediment, with tidal power implications: Severn Estuary, UK. Mar Pollut Bull 2010;61:21–36.
[35] Hill MI, Carr OJ, Birch SP, Parker DM. Cardiff Bay Barrage: environmental assessment and impacts o the natural environment. In: Burt N, Watts J, editors. Barrages: Engineering Design and Environmental Impacts. UK: John Wiley and Sons Ltd; 1996. p. 209–18.
[36] Evans DM, Rogers AP. The effect of the Tawe Barrage on estuarine water quality. In: Burt N, Watts J, editors. Barrages: Engineering Design and Environmental Impacts. UK: John Wiley and Sons Ltd; 1996. p. 301–14.
[37] Shackley S, Dyrynda P. Construction of a tidal amenity barrage within the Tawe Estuary (S. Wales), Impacts on the physical environment. In: Burt N, Watts J, editors. Barrages: Engineering Design and Environmental Impacts. UK: John Wiley and Sons Ltd; 2006. p. 381–94.
[38] Forstner U, Wittmann GTW. Metal Pollution in the Aquatic Environment. Berlin: Springer; 1981.
[39] Kapustka LA, Clements WH, Ziccardi L, Paquin PR, Sprenger M, Wall D. Ecological Effect of Metals. United States Environmental Protection Agency Risk; 2004. Assessment Forum: http://cfpub.epa.gov/ncea/cfm (accessed 10.12.05).

[40] Moriarty F. Ecotoxicology: the Study of Pollutants in Ecosystems. London: Academic Press; 1999.
[41] Clarke RB. Marine Pollution. Oxford: Oxford University Press; 2001.
[42] Spencer KL. Spatial variability of metals in the inter-tidal sediments of the Medway Estuary, Kent, UK. Mar Pollut Bull 2002;44:933–44.
[43] Filgueiras AV, Lavilla I, Bendicho C. Chemical sequential extraction for metal partitioning in environmental solid samples. J Environ Qual 2002;4:823–57.
[44] Turner A, Millward GE. Suspended particles: their role in estuarine biogeochemical cycles. Estuar Coast Shelf Sci 2002;55:857–83.
[45] Couceiro F, Rauen WB, Millward GE, Lin B, Turner A, Falconer RA. Transport and reactivity of nickel in estuarine sediments: results from a high capacity flume. Mar Chem 2009;117:71–6.
[46] Eggleton J, Thomas KV. A review of factors affecting the release and bioavailability of contaminants during sediment disturbance events. Environ Int 2004;30:973–80.
[47] Caetano M, Madureira MJ, Vale C. Metal remobilisation during resuspension of anoxic contaminated sediment: short-term laboratory study. Water Air Soil Pollut 2002;143:23–40.
[48] Saulinier I, Mucci A. Trace metal remobilisation following the resuspension of estuarine sediments: Saguenay Fjord, Canada. Appl Geochem 2000;15:191–210.
[49] Huggett D. The Habitat Directive: a barrier to barrages. In: Burt N, Watts J, editors. Barrages: Engineering Design and Environmental Impacts. UK: John Wiley and Sons Ltd; 1996. p. 247–64.
[50] Carroll J, Boisson F, Fowler SW, Teyssie JL. Radionuclide adsorption to sediment from nuclear waste dumping sites in the Kara Sea. Mar Pollut Bull 1997;35:296–304.
[51] Turner A, Millward GE, Bale AJ, Morris AW. Application of the KD concept to the study of trace metal removal and desorption during estuarine mixing. Estuar Coast Shelf Sci 1993;36:1–13.
[52] Murdoch N, Jonas PJC, Falconer RA, Lin B. A modelling assessment of contaminant distributions in the Severn Estuary. Mar Pollut Bull 2010;61:124–31.
[53] Little DI, Smith J. Appraisal of contaminants in sediments of the Inner Bristol Channel and Severn Estuary. Biol J Linn Soc 1994;51:55–69.
[54] Paalman MAA, Vanderweijden CH, Loch JPG. Sorption of Cadmium on Suspended Matter under Estuarine Conditions – Competition and Complexation with Major Sea-Water Ions. Water Air Soil Pollut 1994;73:49–60.
[55] Millward GE, Liu YP. Modelling metal desorption kinetics in estuaries. Sci Total Environ 2003;314–316:613–23.
[56] McLusky DS. The Estuarine Ecosystem. second ed. Glasgow: Blackie and Sons Ltd; 1989.
[57] Morris JT, Sundareshwar PV, Koepfler EK, Fornwalt B. Phosphorus limitation of coastal ecosystem processes. Science 2003;299:563–5.
[58] Kocum E, Underwood GJC, Nedwell DB. Simultaneous measurement of phytoplanktonic primary production, nutrient and light availability along a turbid, eutrophic UK east coast estuary (the Colne Estuary). Mar Ecol Prog Ser 2002;231:1–12.
[59] Nedwell DB, Dong LF, Sage A, Underwood GJC. Variations of the Nutrients Loads to the Mainland U.K. Estuaries: correlation with Catchment Areas. Urbanization and Coastal Eutrophication. Estuar Coast Shelf Sci 1999;54:951–70.
[60] Berelson WM, Heggie DT, Longmore AR, Kilgore T, Nickolson G, Skyring GW. Benthic nutrient recycling in Port Phillip Bay, Australia. Estuar Coast Shelf Sci 1998;56:917–34.
[61] Casselman ME, Patrick WH, DeLaune RD. Nitrogen fixation in a Gulf Coast salt marsh. Soil Sci Soc Am J 1981;45:51–6.
[62] Goulding KWT. Nitrogen deposition to land from the atmosphere. Soil Use Manage 1990;6:61–6.
[63] Rosenfeld JK. Ammonium adsorption in nearshore anoxic sediments. Limnol Oceanogr 1979;24:356–64.
[64] Yoshinari T. Emissions of N_2O from various environments – the use of stable isotope composition of N_2O as a tracer for the studies of N_2O biogeochemical cycling. In: Revsbech NPS, editor. Denitrification in Soil and Sediment. New York: Plenum Press; 1990. p. 129–50.
[65] Asmus RM, Jensen MH, Jensen KM, Kristensen E, Asmus H, Wille A. The role of water movement and spatial scaling for measurement of dissolved inorganic nitrogen fluxes in intertidal sediments. Estuar Coast Shelf Sci 1998;46:221–32.
[66] Connell DW. Basic Concepts of Environmental Chemistry. Boca Raton, FL: CRC Press; 2005.
[67] Cloern JE. Turbidity as a control on phytoplankton biomass and productivity in estuaries. Cont Shelf Res 1987;7:1367–81.
[68] Hinga KR. Effects of pH on coastal marine phytoplankton. Mar Ecol Prog Ser 2002;238:281–300.
[69] Eyre BD, Ferguson AJP. Comparison of carbon production and decomposition, benthic nutrient fluxes and denitrification in seagrass, phytoplankton, benthic microalgae- and macroalgae-dominated in warm-temperate Australian lagoons. Mar Ecol Prog Ser 2002;229:43–59.
[70] Houghton JT, Callander BA, Varney SK. The Supplementary Report to the IPCC Scientific Assessment. Cambridge: Intergovernmental Panel on Climate Change; 1992.
[71] Jonas PJC, Millward GE. Metals and nutrients in the Severn Estuary and Bristol Channel: contemporary inputs and distributions. Mar Pollut Bull 2010;61:52–67.
[72] Underwood GJC. Microphytobenthos and phytoplankton in the Severn Estuary UK: present situation and possible consequences of a tidal energy barrage. Mar Pollut Bull 2010;61:83–91.
[73] Radford PJ. The impact of a barrage on water quality and ecosystem performance. In: Proc. on Conference on Barrages and Coastal Technology. 1987. p. 18–33.
[74] Valiela I, Alber M, Lmontagne M. Faecal coliform loadings and stocks in Buttermilk Bay, Massachusetts, USA, and management implications. Environ Manage 1991;15:659–74.
[75] Crabill C, Donald R, Snelling J, Foust R, Southam G. The impact of sediment faecal coliform reservoirs on seasonal water quality in Oak Creek, Arizona. Water Res 1999;33:2163–71.
[76] Crowther J, Kay D, Wyer MD. Relationships between microbial water quality and environmental conditions in coastal recreational waters; the Fylde Coast, UK. Water Res 2001;35:4029–38.
[77] Gao G, Falconer RA, Lin B. Numerical modelling sediment-bacteria interaction processes in the Severn Estuary. J Water Resour Prot 2011;3:22–31, http://www.scirp.org/journal/jwarp.
[78] Alkan U, Elliott DJ, Evison LM. Survival of enteric bacteria in relations to simulated solar radiation and other environmental factors in marine waters. Water Res 1995;29:2071–81.
[79] Kay D, Stapleton CM, Wyer MD, McDonald AT, Crowther J, Paul N, Jones K, Francis C, Watkins J, Wilkinson J, Humphrey N, Lin B, Yang L, Falconer RA, Gardner S. Decay of intestinal enterococci concentrations in high-energy estuarine and coastal waters: towards real-time T90 values for modelling faecal indicators in recreational waters. Water Res 2005;39:655–67.
[80] Falconer RA, Lin B, Kashefipour SM. Modelling water quality processes in estuaries. In: Bates PD, Lane SN, Ferguson RI, editors. Computational Fluid Dynamics: Applications in Environmental Hydraulics. UK: John Wiley and Sons Ltd; 2005. p. 304–28.
[81] Falconer RA, Chen Y. Modelling sediment transport and water quality processes on tidal floodplains. In: Anderson MG, Walling DE, Bates PD, editors. Floodplain Processes. Chichester: John Wiley and Sons Ltd; 1996. p. 361–98.
[82] Brown LC, Barnwell TO. The Enhanced Stream Water Quality Models QUAL2E and QUAL2E-UNCAS: Documentation and User Manual. GA: Environment Research Laboratory, US Environment Protection Agency; 1987. Report No EPA/600/3-87/007.
[83] Cerco CF, Cole T. User's Guide to the CE-QUAL-ICM: Three-Dimensional Eutrophication Model. Vicksburg, MS: US Army Corps of Engineers; 1995. Technical Report EL-95-1-5.
[84] Falconer RA, Lin B. Depth Integrated Velocity and Solute Transport (DIVAST) Model, Reference Manual. Cardiff, UK: Hydro-Environmental Research Centre School of Engineering, Cardiff University; 2001.
[85] Saraiva S, Pina P, Martins F, Santos M, Braunschweig F, Neves R. Modelling the influence of nutrient loads on Portuguese estuaries. Hydrobiologia 2007;587:5–18.
[86] Wool TA, Ambrose R, Martin JL. Water Quality Analysis Simulation Program (WASP) Version 6 User's Guide. Atlanta, GA: US Environmental Protection Agency; 2001.
[87] Park K, Kuo AY, Shen J, Hamrick JM. A Three Dimensional Hydrodynamic Eutrophication Model: Description of Water Quality and Sediment Process Submodels, EFDC Water Quality Manual, Report No 327. Gloucester Point, VA: School of Marine Science, Virginia Institute of Marine Science; 2000.
[88] Hipsey MR, Antenucci JP, Hamilton D. Computational Aquatic Ecosystem Dynamics Model: CAEDYM v3, Science Manual. Perth, Australia: University of Western Australia; 2003.
[89] Millero F, Huang F, Zhu X, Liu X, Zhang JZ. Adsorption and desorption of phosphate on calcite and aragonite in seawater. Aquat Geochem 2001;7:33–56.
[90] Bubba MD, Arias CA, Brix H. Phosphorus adsorption maximum of sands for use as media in subsurface flow constructed reed beds as measured by the Langmuir isotherm. Water Res 2003;37:3390–400.
[91] Xia JQ, Falconer RA, Lin B. Hydrodynamic impact of a tidal barrage in the Severn Estuary, UK. Renew Energy 2010;35:1455–68.
[92] R. Ahmadian, Modelling hydrodynamic and bacterial processes through linked river, floodplain and coastal systems: with particular application to marine turbines, barrages and embankment breaches, 2010, unpublished PhD thesis.